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QUANTIFICATION OF BIOLOGICAL INTEGRITY

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Biological integrity may be defined as the maintenance of the community structure and function characteristic of a particular locale or deemed satisfactory to society. In this regard, two points deserve particular emphasis. First, the assumption is made that all natural systems are dynamic and as a result are characterized by a continual succession of species, although the rate of succession may be quite different in different systems. Thus, the protection of a particular species, however valuable from a monetary or other value system, may be counterproductive because it attempts to "freeze" a dynamic system. The conflict between the protection of systems and the protection of species is discussed at length in Cairns (1975).

Second, standards based on this definition of biological integrity will be highly site specific. Therefore, while the criteria (namely maintenance of normal structure and function) will be the same throughout the United States, thus maintaining the equality before the law philosophy, the standards for each particular locale, even within a single state, may be different. This merely recognizes something that the Department of Agriculture has recognized since its inception, i.e., ecological conditions are not the same throughout the United States and attempts to ignore unique regional ecological conditions are stupid and doomed to failure. This would hardly be worth saying were it not for the fact that environmental legislation continues to ignore regional differences probably because of fear of the complexity of the legislation which takes these into consideration. However, it is almost certain that environmental quality control will be unsuccessful until regional differences are acknowledged.

STRUCTURAL INTEGRITY

It is not my intention to attempt to discuss at length the literature showing that natural communities have certain structural characteristics which may be depicted numerically. However, we are all deeply indebted to such early investigators as Preston (1948), Patrick (1949), and others. Also worthy of note is the equilibrium model of MacArthur and Wilson (1963) which showed how community structure could be maintained despite the successional process. In addition to the scientific justification for the use of structural integrity as a means of assessing pollutional changes, there is also substantial benefit in communicability of results, since numbers are more easily understood by nonbiologists than an array of Latin names.

Only three basic kinds of information are presently useful in the quantification of the structural aspects of biological integrity. These are (1) the number of species or other taxonomic units present, (2) the number of individuals per species, and (3) the kinds of species present. Within this framework are such things as spatial relationships, density relationships, and various trophic relationships.

INDICATOR SPECIES

Biologists have long recognized that certain species tend to be found in certain habitats and, therefore, the presence of a certain species indicates that certain ecological conditions exist and the absence of the species that these conditions do not (assuming the species are able to get there). This is, of course, an oversimplification but space does not permit a more detailed discussion. It was probably inevitable that there would be an attempt to transfer this reasoning to the assessment of pollution by stating that certain species are found where polluted conditions exist, others where conditions are semi-polluted, and still another group of species where healthy conditions exist. Unfortunately, pollution covers a much broader range of conditions than most habitats for which biologists predict with reasonable certainty that certain species will be present.

One of the principal faults with the indicator species concept is that a species may be very sensitive to thermal discharges but rather tolerant of a particular chemical toxicant or high concentration of suspended solids, yet all are forms of pollution. A more extended discussion of these weaknesses in the indicator species concept may be found in Cairns (1974a).

However, if one has faith in the indicator species concept, one may collect the species from a particular habitat or locale exposed to a presumed pollutant and determine the number of species in each of the various saprobic categories and either compare these to a reference area not exposed to the source of pollution or some other reference aggregation of organisms. Using this approach, it is possible to get a quantitative comparison of one area with another area. Proponents of the saprobic system have provided increasingly sophisticated analyses of the tolerance of various types of organisms.

Although I do not believe that sufficient information now exists for most areas of the world to make the saprobic system functional, it does seem possible that eventually a sufficiently large information base will revitalize this assessment method. This information base should include detailed information about the tolerance of each species as well as a sufficiently large list of species to insure that an appreciable number will be found in each and every locale where assessments of biological integrity might be made. Until the information base is broader than it is now, the saprobic system does not have general applicability. Since one probably will not know precisely what species are in a particular area until they are collected, much valuable time would be lost if, after collection and identification were completed, one found no saprobic designation for most of the species collected.

THE PATRICK HISTOGRAMS

The histograms developed on the Conestoga survey by Ruth Patrick (1949) and her colleagues at the Philadelphia Academy of Natural Sciences represented a major breakthrough in the quantification of the structural aspects of biological integrity. The principal advantages of this method were: (1) it displayed the number of species in each of seven major categories graphically; (2) it provided a crude means of distinguishing between "normal" abundance and "over" abundance; (3) it permitted detection of gross pollutional effects; and (4) its effectiveness was not markedly reduced by successional changes or small differences in habitat, which would be important if one were using species lists alone.

The principal weaknesses of this method were: (1) a highly trained team of specialists in different taxonomic disciplines was required and thus the method was difficult to use on a broad scale because of the lack of skilled specialists; (2) it only provided four major categories of "health" and very often the presence or absence of a few species might alter the designation from one category to another (this, of course, could be offset by an extended discussion which would complicate the communication problem); (3) the time required to obtain the information often extended to months because of the difficulty of identifying certain species (of course, in an emergency situation this could be substantially shortened, but nevertheless, the identification process requires at least several weeks). One should remember that this method was developed over 26 years ago and it should be judged in the context of its time—at that time it represented a major turning point in the quantification of biological integrity.

A number of methods followed which attempted to reduce the number of the specialists required and the complexity of the Patrick histograms and retain the basic analytical thrust. Examples of these are the methods of Beck (1954, 1955) and Wurtz (1955). These latter methods combined elements of the saprobic system with the Patrick method and concentrated on a relatively narrow spectrum of the aquatic community in order to simplify identification and analytical problems. It is probably fair to say that they represented a variation on already established themes and not a conceptual advance. The work of Gaufin and Tarzwell (1952) and Gaufin (1956) on Lytle Creek, based on the same assumptions as Patrick's, represented a major contribution in the quantification of biological integrity since it showed the quantitative and qualitative structural changes that occurred when an aquatic community had been severely stressed by pollution and underwent a recovery process.

BEAK METHOD

The method developed by Beak, et al. (1959) was primarily for lakes but might well work in certain streams where there are one or two species persisting for a substantial period of time in substantial numbers. Essentially the method consisted of determining the density of one or more established species in two concentric rings at different distances from the waste outfall. Changes in proportional abundance in these two rings indicated pollution since presumably there would be a concentration gradient proceeding away from the outfall in much the same manner that ripples expand as they leave the spot where a thrown stone enters pond water.

There are several advantages to this method: (1) one does not need a substantial amount of taxonomic expertise because only a few species are involved; (2) the results are expressed on a graded scale; and (3) the method provides for determining with 95 to 99 percent confidence whether or not a significant change occurred.

Among the disadvantages are: (1) the chance that the organism or organisms one has selected may be highly resistant to the particular stress being assessed; (2) the "noise factor" in density assessments is often quite high; (3) one's test species may be wiped out by some natural catastrophe and leave one without any way of determining whether or not pollutional stress has occurred.

PATRICK DIATOMETER

The diatometer developed by Ruth Patrick, Matthew Hohn, and John Wallace (1954) represented a substantial advance in the quantification of the structural component of biological integrity because with the use of an artificial substrate, it substantially reduced the "noise factor" due to habitat differences and time of substrate exposure. In addition, it used a more sophisticated method based on a log-normal distribution of species abundance developed by Preston (1948).

Preston (1948) showed that for a sufficiently large aggregation of individuals of many species, the species-abundance relationship often conformed to a normal law, after the individuals were grouped on a logarithmic scale. That is, the observed distribution could be graduated by

$$y = y_o \exp - (aR)^2 \qquad (1)$$

where y represents the number of species falling in the Rth "octave" to the left or right of the mode, y_o is the number of species in the modal octave, and "a" is a constant that is related to the logarithmic standard deviation, σ, by

$$a^2 = 1/2\sigma^2 \qquad (2)$$

Preston's original method involved grouping the individuals into octaves with end points r = 1, 2, 4, 8 . . . These end points were subsequently labeled 1 through R, the total number of octaves. Those species that fell on a group end point were split equally between that octave and the next higher or lower octave. If an entire log-normal population is censused the curve extends infinitely far to the left and right of the mode and is symmetrical. As Preston (1962) pointed out, however, species are not found infinitely far from the mode in either direction and he describes an intuitively reasonable method of determining the end of the real finite distribution of individuals and species.

Given then, a complete ensemble or universe, the nature of the distribution can be ascertained. However, it is exceptional in ecological work that a complete universe, or "population," or "community," et cetera is fully censused and in most instances one must be content to deal with samples from a universe (Preston, 1962). Provided that a sufficiently large random sample can be drawn from the universe, the distribution will be truncated on the left, indicating that there are additional, uncensused species in the ensemble, although they may comprise only a relatively small percent of the total. To census these species (i.e., to obtain the universe) would require extraordinarily large collections which, for practical purposes, would be out of the question. However, provided that the sample is large enough to ascertain the mode of the distribution, both y_o and σ can be determined and thus, the extent of the complete, untruncated log-normal distribution (i.e., the number of species in the universe). Deducing the universe from a random sample is carried out by means of internal evidence at hand and not by an external assumption; the universe we deduce is based on the nature of our sample (Preston, 1962).

Species-abundance relationships are based upon two fundamental types of data: the number of species in the community or universe and the relative proportions of individuals among the species. For purposes of quantifying these relationships it is advantageous for the ecologist to be able to summarize his data in one or two descriptive "community statistics." When the data conform to the log-normal distribution, the obvious descriptors would be the logarithmic variance, σ^2, and N, the number of species in the biological universe. Other suitable parameters are *a*, a measure of dispersion, and y_o, the height of the mode. The important point is, however, that the species-abundance relationship can be adequately summarized by one or two general parameters which facilitates quantitative comparison of two or more communities.

A number of difficulties arise if the log-normal distribution is relied upon as the underlying theoretical relationship of species abundance. First, it has not been shown to be sufficiently widely applicable to all types of biological ensembles, to date. Preston (1962) cites numerous cases where the log-normal is adequate and Patrick, et al. (1954) have used this distribution to describe the occurrence of diatom species in fresh and brackish water environments. However, these are limited applications of the theory and do not confirm its ecological universality. Biological ensembles of relatively small extent usually require other methods of quantification since their distribution cannot be graphed with much success.

Secondly, a large amount of data must be collected and censused, even for the truncated form, so that the mode of the distribution can be exposed. In some situations, this fact alone prohibits the use of the log-normal distribution. Thirdly, the estimation of the parameters of the distribution, i.e., the mean and standard deviation, is difficult although computer programs for this purpose are available (Stauffer and Slocomb, manuscript in prep.).

These are the primary reasons that ecologists have turned away from this type of species-abundance quantification in favor of methods that do not depend upon the theoretical form of distribution of individuals among species. Indices based on information theory, although more difficult to visualize biologically, have gained great popularity as descriptive measures of community structure. Despite its drawbacks the diatometer method is one of the soundest available for the quantification of biological integrity. The aquatic ecology group at Virginia Tech has a substantial program designed to reduce some of these problems (Cairns, et al. 1974; Stauffer and Slocomb, manuscript in prep.). This indicates our belief that the method is basically sound and will continue to provide valuable information about biological integrity.

DIVERSITY INDICES

The diversity index is probably the best single means of assessing biological integrity in freshwater streams and rivers. It is less effective and may even be inappropriate in lakes and oceans. As a screening method for locating trouble spots in most flowing systems, it is superb! Unfortunately, many investigators looking for a single all-purpose method, use it alone when an array of evidence is required. Beware of the investigator who tries to use a single line of evidence of any type instead of multiple lines of evidence to assess biological integrity. A brief discussion of diversity indices follows.

Diversity indices that permit the summarization of large amounts of information about the numbers and kinds of organisms have begun to replace the long descriptive lists common to early pollution survey work. These diversity indices result in a numerical expression that can be used to make comparisons between communities or organisms. Some of these have been developed to express the relationships of numbers of species in various communities and overlap of species between communities.

The Jaccard Index (1908) is one of the most commonly used to express species "overlap." Other indices such as the Shannon-Weiner function (Shannon and Weaver, 1963) have been used to express the evenness of distribution of individuals in species composing a community. The diversity index increases as evenness increases (Margalef, 1958; Hairston, 1959; MacArthur and MacArthur, 1961; and MacArthur, 1964). Various methods have been developed for comparing the diversity of communities and for determining the relationship of the actual diversity to the maximum or minimum diversity that might occur within a given number of species. Methods have been thoroughly discussed by Lloyd and Ghelardi (1964); Patten (1962); MacArthur (1965); Pierou (1966, 1969); McIntosh (1967); Mathis (1965); Wilhm (1965) and Wilhm and Dorris (1968) as to what indices are appropriate for what kinds of samples. An index for diversity of community structure also has been developed by Cairns, et al. (1968) and Cairns and Dickson (1971) based on a modification of the sign test and theory of runs of Dixon and Massey (1951).

Diversity indices derived from information theory were first used by Margalef (1958) to analyze natural communities. This technique equates diversity with information. Maximum diversity, and thus maximum information, exists in a community of organisms when each individual belongs to a different species. Minimum diversity (or high redundancy) exists when all individuals belong to the same species. Thus, mathematical expressions can be used for diversity and redundancy that describe community structure.

As pointed out by Wilhm and Dorris (1968) and Patrick, et al. (1954), natural biotic communities typically are characterized by the presence of a few species with many individuals and many species with a few individuals. An unfavorable limiting factor such as pollution results in detectable changes in community structure. As it relates to information theory, more information (diversity) is contained in a natural community than in a polluted community. A polluted system is simplified and those species that survive encounter less competition and therefore may increase in numbers. Redundancy in this case is high, because the probability that an individual belongs to a species previously recognized is increased and the amount of information per individual is reduced.

The relative value of using indices or models to interpret data depends upon the information sought. To see the relative distribution of population sizes among species, a model is often more illuminating than an index. To determine information for a number of different kinds of communities, diversity indices are more appropriate. Many indices overemphasize the dominance of one or a few species and thus it is often difficult to determine, as in the use of the Shannon-Weiner information theory, the difference between a community composed of one or two dominants and a few rare species, or one composed of one or two dominants and one or two rare species. Under such conditions, an index such as that discussed by Fisher, Corbet and Williams (1943) is more appropriate.

FUNCTIONAL INTEGRITY

Only a limited effort has been made to assess the impact of pollutional stress on the functioning of aquatic communities. Nevertheless, it has become increasingly evident that approaches and methods to evaluate the effects of stress on the functioning of aquatic communities are badly needed. Functional characteristics of aquatic ecosystems such as production, respiration, energy flow, degradation, nutrient cycling, invasion rates, et cetera are related to the activities of various components of the aquatic community. The importance of these activities is obvious yet the availability of methods of studying these activities is miniscule.

The importance of being able to evaluate the effects of pollutants on both the structure and function of aquatic communities has been recognized by the Institute of Ecology's Advisory Group to the National Commission on Environmental Quality which has identified biological integrity as the pivotal issue in the assessment of pollution effects. Their definition of biological integrity (which this author helped prepare) emphasizes both the structural and functional aspects of natural ecosystems and communities. In addition, The Federal Water Pollution Control Act Amendments (PL 92-500, Sec. 304) states water quality criteria should reflect the latest scientific knowledge on the effect of pollutants on biological community diversity, productivity, and stability, including information on the factors affecting rates of eutrophication and rates of organic and inorganic sedimentation for various types of receiving water.

In general, assessing the impact of pollution on aquatic systems has been troublesome. Community structure analysis has been preferred to investigations of function in dealing with perturbation of aquatic systems because its study is less time consuming, better understood, requires less effort, and has become conventional. Community function has been avoided because methods dealing with its complex operation have been lacking. Furthermore, field studies have been hindered by fluctuating environmental quality plus the fact that the dynamics of systems are inherently more difficult to measure than the components themselves. Clearly, there is a need for the study of aquatic community structure. However, such studies provide incomplete information. Function must be coupled with community structure investigations to obtain a full understanding of the effects of pollution relative to the health of aquatic communities.

Structural analyses in the form of species diversity, species lists, and numbers of organisms have not adequately filled the regulatory agency's need for information on the response of aquatic communities to pollutional stress. Diversity indices place values not on organisms which may be present in small numbers but on those which perform vital functions in the maintenance of community integrity. The symptoms of pollution may be masked by shifts in the dominance of some community members without substantially altering the diversity. It is also difficult to establish whether these shifts or changes are beneficial, detrimental, or indifferent. Because identical assemblages of organisms never reoccur in natural systems, there is no "true natural fauna" which remains constant through time. The high functional redundancy of communities makes it possible to lose one or several pollution-sensitive species and still maintain adequate function. Species lists and numbers give little information other than what is present in an aquatic community at any point in time.

Function, however, provides better insight into the interaction of populations, the cycling of energy, and nutrient exchange in a community. Ideally, any studies of communities affected by pollution should include both structural and functional assessment, as well as the possible interrelationships between the two components.

Although there is no body of quantitative methods for the assessment of the functional integrity of biological systems there are a number of possibilities. A few examples of these follow (if I have left out your favorite method, don't write to me, use your energy to perfect its application in the determination of functional integrity).

PROTOZOAN INVASION RATES

It is possible that a determination of the invasion rate of a protozoan-free substrate placed in a freshwater lake or stream may alone be sufficient to estimate the degree of eutrophication; if this is the case, a rather easily carried out assessment requiring only a few days will be available. It is also highly probable, however, that additional useful information will be gained from determining the time to reach equilibrium even though this may require 2 or 3 months in some cases.

Since 1966, Cairns and a number of associates, (principally Dr. William H. Yongue, Jr.) have been carrying out investigations involving the colonization of polyurethane foam anchored in various portions of Douglas Lake, Mich., (e.g., Cairns, et al. 1969, 1973; Cairns and Yongue, 1974) and other sections of the country ranging as far south as South Carolina (Yongue and Cairns, 1971). The purpose was to study the MacArthur-Wilson Equilibrium Model of the point at which the colonization rate is in rough equilibrium with the decolonization rate. A few years ago, when looking over the assembled data from 1966 through 1972, it became evident that the time required to reach an equilibrium between these two rates had been steadily decreasing in Douglas Lake, Mich., from an initial period of approximately 8 weeks to a period of approximately 2 weeks in 1974.

Examination of colonization rates and the time required to reach equilibrium from other locations strongly suggested a correlation between the degree of eutrophication of the lake or pond in question and the time required to reach equilibrium. Cairns (1965), in a year-long study of the Conestoga Basin carried out in 1948, noted that the initial response of a freshwater protozoan community to increased nutrient loading was to increase both the number of species and the number of individuals per species. Further increase in nutrients might then lead to a decline in the number of species, but not necessarily the number of individuals. This has subsequently been confirmed in a number of situations.

ZOOPLANKTON PHYSIOLOGY AND REPRODUCTION

In a time of increased power demands, more and more power plants are being constructed. Since large volumes of water are used for cooling these plants considerable attention has been focused on their effect on aquatic populations, especially fish. Very little work has been conducted on zooplankton, and many of these papers do not examine the interaction of temperature changes, chlorine, and physical damage. Most of these studies are only on acute mortality. Chronic studies are virtually nonexistent (Bunting, 1974). Various methods have been proposed to examine the effects of entrainment but no studies have been done to determine effectiveness and interrelatedness of the methods.

Such parameters as zooplankton physiology and reproduction might be useful in estimating the functional integrity of zooplankters in lakes near power plants. For example, oxygen consumption rates, ATP, and lipid concentrations could be changed. Filtering rate of zooplankters might also be determined by comparing algal counts at time zero and after a defined period of time (Buikema, 1973a) using the equation

$$F.R. = v\frac{\log_{10}c_0 - \log_{10}c_t}{\log_{10}e}$$

These factors all affect reproduction rates (Buikema, 1973b) which could be quantitatively assessed in a relatively short period of time. Because many zooplankters migrate vertically in response to changes in light intensity, such functional assessments as rates of migration (Gehrs, 1974), response thresholds, et cetera could be useful.

FUNCTIONING OF BENTHIC MACROINVERTEBRATE COMMUNITIES

Methods for the assessment of benthic macroinvertebrate community function have unfortunately lagged far behind the development of those for the analysis of community structure.

A great body of literature has been amassed recently, firmly establishing the energy supply of many running water systems as largely heterotrophic (Minshall, 1967, 1968; Vannote, 1970; Fisher, 1971; Hall, 1971; Kaushik and Hynes, 1968; Fisher and Likens, 1972; Cummins, 1972, 1973; Cummins, et al. 1973a, 1973b). Detrital-based ecosystems have been shown to be largely dependent upon litter from their terrestrial surroundings for nutrient input and even the evolutionary dispersal of insects (Ross, 1963). The processing of dead organic material passing downstream through a stream ecosystem is largely a function of primary decomposition by fungi and bacteria (Iversen, 1973), and the selective feeding on detritus by invertebrates following microfloral colonization and conditioning (Petersen and Cummins, 1974).

Aquatic macroinvertebrates are important components in food webs of aquatic systems, being primary and secondary consumers, and serving as food sources for higher trophic levels. Very little information exists on the functioning of macroinvertebrate communities, and even less concerning the influence of pollutional stress on feeding patterns of invertebrates. The loss of an individual in a community with a particular feeding pattern due to pollution and its effect on community function have not been investigated. More studies to develop methods for the assessment of macroinvertebrate community function are needed.

Identification of the importance of the major biological components in the process of reducing the initial detrital biomass could be accomplished by artificially selecting against specific components in simultaneous parallel experiments, e.g., according to their size. Evaluation of the importance of each of the components can be based upon a number of parameters including:

a. Standing biomass.
b. Calorific equivalents.
c. Efficiency of energy utilization.
d. Size of food particles required.
e. Specific interrelationships between the components.

If simultaneous parallel experiments are conducted concurrently under experimental (stressed) and control (absence of stress) conditions, patterns of the impact of stress should emerge.

AUTOTROPHIC AND HETEROTROPHIC FUNCTIONING

The autotrophic and heterotrophic components of aquatic systems have a vital and essential role in the regulation of the functional activities of aquatic systems. These two components of aquatic communities are intimately involved in nutrient cycling, energy fixation, and energy transfer. In order to understand and predict the functional capabilities of freshwater flowing systems, it is obvious that methods must be evaluated which allow a better understanding of the above activities.

The energetics of freshwater flowing systems depend upon two sources of carbon—carbon fixed internally via the photosynthetic activity of the autotrophic community and carbon which enters the system from the terrestrial environment (i.e., allochthonous material—leaves, et cetera). The utilization of either of the sources is generally the rate-limiting step in the total energetics of the whole aquatic system. In addition, the availability and utilization of essential inorganic nutrients in freshwater flowing systems are governed to a large extent by the autotrophic and heterotrophic components. While the autotrophic and heterotrophic microbial communities are frequently not studied in evaluating the impact of stress (pollution) on flowing systems, they have been shown to be sensitive and reliable indicators of environmental perturbation (Cairns, 1971). The reason that they have not been utilized extensively in pollution assessment in the past has been directly related to the difficulty in measuring their structure and function. However, it is essential that approaches be developed which allow us to understand and measure the responses of these vital components of aquatic systems.

A. Nitrogen Cycle—Procedures permitting an evaluation of the process and role of the nitrogen cycle in flowing fresh water should be further developed, although some excellent references are available (e.g., Brezonik and Harper, 1969; Klucas, 1969; Kuznestor, 1968 and Tuffey, et al. 1974). The ability of microorganisms to enzymatically transform one chemical species into another is well known. For example CO_2 is reduced to organic compounds and condensed phosphates can be broken down into phosphates and then the phosphates can be coupled to organic molecules to form some extremely important biological molecules. However, nitrogen appears to be utilized in more forms by living organisms than any other element.

Nitrogen can exist in six different valence states and all six of these states can be utilized or produced by microorganisms. In some cases there are specific organisms for specific ionic species such as nitrifying bacteria for NH_3 and NO_2 or nitrogen fixers for N_2. On the other hand a myriad of organisms may use NH_3 or NO_3. Microorganisms are also known to produce NH_3, NO_2, NO_3, N_2 and organic nitrogen compounds. Quite obviously microorganisms play an important role in the nitrogen cycle, and nitrogen plays an important role in the life of living organisms.

The investigation of the nitrogen cycle or balance in a stream or lake is a viable approach for studying functional responses at the microorganism level because: 1) many if not all of the enzymatic reactions involved in the nitrogen cycle are heat sensitive; 2) many of the reactions are sensitive to heavy metals; 3) many are directly affected by oxygen levels; and 4) the analytical methods for investigating the microorganisms involved in the N cycle are fairly well known.

B. Carbon Cycle—Rates of carbon uptake and incorporation are integrally involved with structure and function of autotrophs and heterotrophs in aquatic systems (Patrick, 1973; Bott, 1973). Knowledge of these rates yields information vital to the understanding of carbon cycling and assimilative capacity in diverse freshwater flowing systems which are subjected to pollution (Saunders, 1971; Wetzel and Rich, 1973). Numerous techniques have been developed utilizing radioisotopes and the stoichiometric relationship between oxygen consumed or produced in the system to carbon oxidized or reduced (Vollenweider, 1969; Hobbie, 1971). Enough variation exists in current investigations to make comparisons from investigation to investigation exceedingly difficult. The development of acceptable techniques which are efficient and can be utilized in a variety of situations is overdue. Data collected using a somewhat universal and standard technique or method would be comparable to data collected in the same manner in other investigations. The value of such an approach could be realized in information gained from the statistical comparisons of diverse aquatic systems and pollutional regimes (Steel and Torrie, 1960).

In an effort to develop such methodology for determining the role of carbon in freshwater flowing systems, a variety of old and new techniques might be examined. This could include development of equipment and test chambers, evaluation of the efficacy of neutron activation analyses, autotrophic indices, plant and animal carotenoids, other pigments (xanthophylls, phycobilins, et cetera), ATP analyses, and other procedures (Margalef, 1960; Thatcher and Johnson, 1973). The development of procedures with the little-used isotopes (^{35}S, ^{33}P, et cetera to determine functioning of the heterotrophic and autotrophic components could also be investigated (Bott and Brook, 1970).

C. Sulfur Cycle—One might develop a method to differentiate heterotrophic and autotrophic function via the preferential utilization of organic and inorganic sulfur. Sulfur, as one of the macronutrients and an almost universal component of proteins, may be intimately involved with structure as well as function of flowing aquatic systems. Knowledge of uptake rates and utilization by the heterotrophic and autotrophic components may yield valuable information about reaction rates and critical concentrations in these compartments (Vollenweider, 1969). Sulfur has recently been cited as a critical factor in the acidity of rainfall and runoff, pollution from acid mine drainage, and release or leaching of nutrient cations such as calcium from the soil into aquatic systems (sulfate is the most common inorganic species in flowing systems). Many microorganisms and plants have the capacity to reduce sulfate to the oxidation state of sulfide for incorporation into organic materials such as the amino acids, cysteine and methionine (Rodina, 1972).

The differential metabolism of sulfur by heterotrophs and autotrophs in aquatic systems could be determined using SO_4 and tagged amino acids. Changes in community structure above and below polluting discharges could be monitored. It has been shown that bacterial diversity and algal diversity are reduced by thermal discharges (Guthrie, et al. 1974). Rates of sulfur metabolism above and below thermal effluents may be correlated with changes in diversity and may be directly involved with the assimilative capacity of that reach of the stream.

HISTORIC PROSPECTIVE

Even if we quadrupled the number of methods available for the quantification of biological integrity, a very basic question will not be resolved—namely, what type of system will be used to provide the baseline or reference numbers against which systems receiving industrial waste discharges or other forms of contamination may be compared? The selection process is likely to be hampered by our failure to recognize that most of the continental United States, particularly the area east of the Mississippi, is a man-altered environment. Two illustrations will suffice to make this point.

Each summer a number of students and faculty members (frequently including me) journey to the northern tip of Michigan's lower peninsula to spend a few months studying "natural" ecosystems at the University of Michigan Biological Station. Students and faculty are cautioned not to overcollect and upset the balance of nature. I remember vividly a stormy faculty session many years ago when one faculty member proposed experimental clearcutting on the station tract which was hotly contested by another faculty member who wanted to preserve the "natural environment" for certain plants. The present station director, David M. Gates, who spent his boyhood at the station, remembers from personal observation and from the journals of his plant ecologist father, Frank Gates, the devastation that resulted from the vast lumbering activities characteristic of that area less than 100 years ago. Journals of early biologists report that it was almost impossible to travel anywhere in the area during the logging period without seeing slash fires or smoke from them. Originally the Biological Station area was used by civil engineers because the vegetation had been so thoroughly destroyed that long lines of sight were possible. Only when the vegetation became partially reestablished and interfered with surveying was the area turned over entirely to biologists.

The second example is the area roughly between Allendale, S.C., and Augusta, Ga., known as the Savannah River tract. This site was placed off limits in approximately 1951 by the Atomic Energy Commission. At this time the inhabitants of the town, Ellenton, and the other parts of the area were removed and the farms, homesites, et cetera were mostly allowed to revert to "natural conditions" following a brief occupancy by work crews during the construction phases. Some trees were planted and other assists were given to natural reinvasion, but mostly the process of change was "natural." Today the area abounds with wildlife and is and has been the focus of many ecological surveys and studies. Were it not for the AEC facilities still there, most ecologists would not hesitate to consider this a natural system and for the vast portions of the tract where no manmade structures are present, most ecologists would have no hesitation in carrying out an extended study of a "natural system." The point of all this is that what we are willing to label as "natural" systems often were at one time substantially altered by human activities and frequently underwent severe ecological perturbations before reaching their present state. Thus, we should not hesitate to use as reference systems those we consider satisfactory even if they were once altered by human activities.

ANTHROPOCENTRIC CRITERIA

One might also characterize biological integrity by determining the ability to produce desired quantities of commercially or recreationally desirable species. This could be quantified by comparing actual yield against an optimal yield. This might also be done for pest or nuisance species such as biting insects or algae that produce unpleasant tastes and odors in water supplies. Quantification in terms of aesthetically desirable species boggles the mind.

MANAGEMENT CRITERIA

Using the anthropocentric criteria discussed above, one might also quantify the energy and material required to maintain the desired crops or low densities of pest organisms. One might rate a system from 1 to 10 on whether it provides desirable conditions "free" (i.e., no management costs) or whether it is a costly system to manage. This might be considered an operational definition of biological integrity.

ASSIMILATIVE CAPACITY

For the purposes of this discussion assimilative capacity is defined as the ability of a receiving system or ecosystem to cope with certain concentrations or levels of waste discharges without suffering any significant deleterious effects. I have attempted to find other definitions of assimilative capacity but unfortunately have failed to find any significant body of literature on this subject.

A common position of those denying that assimilative capacity exists is that an introduced material will cause a change that would not have otherwise occurred. There is an implication that any change is necessarily deleterious. It would not be rational to deny that an ecosystem or a community of organisms will respond to an environmental change whether caused by the introduction of wastes from an industrial process or a leaf dropping into the water from a tree. It seems to the detractors of the concept of assimilative capacity that the introduction of chemicals and fibers in the form of a leaf is not regarded as a threat to the biological integrity of the receiving system (i.e., the maintenance of the structure and function of the aquatic community characteristic of that locale) but that the introduction of the same chemicals and fibers via an industrial municipal discharge pipe would be deleterious. Some of the same chemicals and materials present in leaves and other materials introduced naturally into ecosystems are also present in the wastes from industrial and municipal discharge pipes. Surely, within certain limits, an ecosystem can cope with these. Of course, for bioaccumulative materials such as mercury, the assimilative capacity of many receiving systems is almost certainly very low and for some probably zero. Zero discharge of these types of polluting materials is a highly desirable goal which should be achieved with dispatch. For other types of materials the case against assimilative capacity appears to have no logical framework.

The weaknesses of the argument against the use of nondegrading assimilative capacity are especially weak where heated wastewater discharges are concerned. Would a ΔT of 0.001°C damage the biological integrity of the Mississippi River at New Orleans or the Atlantic Ocean near Key West, Fla.? Or to phrase the question somewhat differently, would this thermal addition of industrial origin be more of a threat or even a measurable threat to the biological integrity of these ecosystems than the natural thermal additions? Is a microgram of glycolic acid placed into an ecosystem by an algal population less of a threat to the biological integrity of an ecosystem than the same amount discharged from an industrial or municipal waste pipe? If you believe that there are some ecosystems into which this amount of heat or glycolic acid might be introduced without damaging the structural or functional integrity of the ecosystem, then you cannot deny the existence of assimilative capacity even if there are a very limited number of ecosystems for which you think this is possible.

A second aspect of the assimilative capacity controversy is the belief of some environmentalists that the ecological effects of society's activities, particularly those of industrial origin, can be forever contained. The position of people who proclaim that we can have an industrial society from which nothing may be introduced into ecosystems is irrational. If one's outlook is towards treating each waste discharge in isolation from all others, rather than viewing it regionally, then it is quite evident that the complete treatment of industrial wastes will require substantial amounts of energy, an enormous capital investment in facilities, and a variety of chemicals to facilitate the treatment process if the process water is to be recycled for further use. This energy, chemicals for treatment, and materials for the construction of additional treatment facilities will all have to be produced somewhere and the production process will produce heat and other waste products that cannot be forever contained.

The critical question is not whether they can be introduced into the environment, thus taking advantage of its assimilative capacity, but rather how and where they should be introduced into the environment. If the detractors of the assimilative capacity approach believe that it is possible to have an industrial society without introducing anything into the environment at any place or time then they should show us in a substantive way how this can be done. If they cannot do this, then we should all collectively address the problem of determining the assimilative capacity for different types of waste products and ecosystems rather than endlessly discussing whether assimilative capacity does or does not exist. If the antagonists of the concept of assimilative capacity believe that there is no way in which a harmonious relationship between an industrial society and ecosystems can be achieved then they should tell us in more detail what to do next.

There is no question that the use of assimilative capacity increases the risk of damaging the biological integrity of the receiving system. Even the use of nondegrading assimilative capacity reduces the safety factor and this, together with the variability in assimilative capacity caused by changing environmental conditions, makes it essential that continuous biological monitoring be used by dischargers taking advantage of assimilative capacity. Thus, management and monitoring costs are inevitable but may often be less expensive than advanced waste treatment costs.

RESISTANCE TO AND RECOVERY FROM CHANGE

A. Resistance to Change—Some biological communities have a greater inertia or resistance to disequilibrium than others. The ability to resist displacement of structural and functional characteristics is a major factor in the maintenance of biological integrity. In order for a displacement to be categorized as a loss of integrity it would have to exceed the range of oscillations or fluctuations characteristic of that system. If sufficient funds and time are available these natural fluctuations can be determined with reasonable accuracy. The stress required to overcome inertia is less easily determined until an actual displacement has occurred, although one might make an estimate from dose-response curves of important indigenous organisms. However, the use of the "species of interest or importance to man" concept is dangerous since the response of that species to a particular stress might not be representative of the response of other species in the system. At present we have no reliable means of quantifying this important characteristic of biological integrity. A very crude estimate of inertial rank ordering of major water ecosystems is given in Table 1 (from Cairns, 1974).

Table 1.—The relative elasticity (ability to return to normal after being displaced or placed in disequilibrium), inertia (ability to resist displacement or disequilibrium), and environmental stability (consistency of chemical-physical quality).

	Elasticity	Inertia	Environmental stability
Lakes	2	3	2
Rivers	1	2	3
Estuaries	3	1	4
Oceans	4	4	1

1 = high
2 = intermediate high
3 = intermediate low
4 = low

B. Recovery from Stress—Once a system has been displaced (i.e., altered structurally or functionally) the time required for the restoration of biological integrity is important as are the factors which affect the recovery process. Accidental spills such as the ones reported by Cairns, et al. (1971, 1972, 1973), Crossman, et al. (1973) and Kaesler, et al. (1974) for the Clinch and other rivers will probably always occur in an industrial society. A crude index of elasticity (i.e., ability to snap back after displacement) has been developed by Cairns (in press). A list of the factors important to this index follows:

a. Existence of nearby epicenters (e.g., for rivers, tributaries) for reinvading organisms.

Rating system—one = poor; two = moderate; three = good.

b. Transportability or mobility of disseminules.

Rating system—one = poor; two = moderate; three = good.

c. General present condition of habitat following pollutional stress.

Rating system—one = poor; two = moderate; three = good.

d. Presence of residual toxicants following pollutional stress.

Rating system—one = large amounts; two = moderate amounts; three = none.

e. Chemical-physical water quality following pollutional stress.

Rating system—one = severe disequilibrium; two = partially restored; three = normal.

f. Management or organizational capabilities for immediate and direct control of damaged area.

Rating system—one = none; two = some; three = thriving with strong enforcement prerogatives.

Using the characteristics listed above, which must be placed into the equation in exactly the sequence in which they are given, one can arrive at a rather crude approximation of the probability of relatively rapid recovery. This would mean that somewhere between 40 and 60 percent of the species might become reestablished under optimal conditions in the first year following a severe stress, between 60 and 80 percent in the following year, and perhaps as many as 95 percent of the species by the third year. Natural processes with essentially no assistance from a management or a river basin group accomplished this on the Clinch River spills which were studied by the Aquatic Ecology Group at Virginia Tech and the usefulness of this estimate has also been checked with data provided by some acid mine drainage studies (Herricks and Cairns, 1972, 1974a, 1974b) and seems adequate in this regard as well. The equation follows:

RECOVERY INDEX = a x b x c x d x e x f
400+ = chances of rapid recovery excellent
55-399 = chances of rapid recovery fair to good
less than 55 = chances of rapid recovery poor

During the development of the simplistic equation just given, considerably more complicated equations were considered and rejected because the refinements seemed meaningless in view of our present state of knowledge. On this basis one might reject even the modest effort just made. On the other hand there seems to be a very definite need to formalize the estimation of recovery and one hopes that more precise equations properly weighted will evolve from this modest beginning.

CONCLUSIONS

It is evident that no single method will adequately assess biological integrity nor will any fixed array of methods be equally adequate for the diverse array of water ecosystems. The quantification of biological integrity requires a mix of assessment methods suited for a specific site and problem (e.g., heated wastewater discharge). Since some ecosystems are more complex than others and some stresses on biological integrity more severe than others the variety and intensity of methods used should be site specific. Table 2 demonstrates a simple version of a decision matrix for resolving these problems.

Table 2.—Potential threat to biological integrity.

Ecosystem Complexity	Minor 1	Moderate 2	Serious 3
Simple 1	1	2	3
Intermediate 2	2	4	6
Complex 3	3	6	9

A number one situation would require less effort and fewer criteria than a number nine. Many ecologists will be unwilling to make the value judgments necessary for even the simple example matrix. Unless they dispute the assumptions which preceded the matrix, professional pride should force them to do so because otherwise they will be forcing industry to overassess as a result of their insecurity and inability to make distinctions between difficult and simple situations.

What is needed is a protocol indicating the way in which one should determine the mix of methods that should be used to estimate and monitor threats to biological integrity. A good example of this approach is "Principles for Evaluating Chemicals in the Environment" (originally "Principles of Protocols for Introducing New Chemicals into the Environment") published in 1975 by the Environmental Studies Board of the National Academy of Sciences. A badly needed accompaniment is a national system for storing data gathered for such purposes which also insures greater standardization and compatibility than is possible with present systems. It is also important that industrially sponsored studies of this kind be made more accessible to the academic community. This would insure that shoddy contractors would be exposed by academic criticism and reduce the money wasted by industry on these groups and would also expedite advancement of this type of assessment which would also benefit industry.

While additional methods for quantifying biological integrity are being developed, industry and other dischargers into aquatic systems can take immediate measures to protect ecosystems. Until all currently available methods have been used there is no justification for complaining about the lack of appropriate methodology. A list of some useful methods follows: (1) A screening test such as the ORSANCO 24-hour test to determine which wastes require immediate attention and which may be relegated to a lower priority. (2) A determination using the ORSANCO 24-hour or some other short term test of the variability of waste toxicity. Although there is some variability in bioassays due to the nature of the test organisms, it is dwarfed by the variability in toxicity of most industrial wastes. Some wastes may vary in toxicity as much as 10,000 times from one sample to another. (3) A baseline ecological survey which is essentially an inventory of biological, chemical, and physical conditions in the receiving system, should be carried out at critical points related to the discharge with, of course, an unexposed area to serve as a reference or control. (4) A biological monitoring of certain areas of the receiving system on a routine basis so that any deleterious effect can be determined rapidly.

There exists a vast array of methods potentially suitable for the assessment or determination of biological integrity, both structural and functional. One generally, though not invariably, realizes the importance of the biological entity being measured, although even this is not always the case. However, showing that it is useful in the context of measuring changes in biological integrity is another matter. This is where biologists and ecologists have usually dropped the ball.

Most biologists and ecologists with classical training do not feel any responsibility to justify the appropriateness of a method or a parameter being suggested as a monitor for biological integrity, because they assume that if it is useful in ecology, and if classical ecologists recognize the importance of the measurement, then it must be appropriate for the assessment of biological integrity. We have all seen the endless "shopping lists" resulting from a group of ecologists putting together a list of parameters to determine the ecological impact of a particular activity such as the construction of a dam. The methods often appear to be assembled by a "stream of consciousness process," or each of the ecologists present flushes his or her mind of all the methods known to him and requests that determinations be made.

Even when all this information is collected, classical ecologists will often refuse to predict the consequences of the course of action anyway because the information is often not gathered in an orchestrated fashion so that the data bits can be integrated and correlated. On such projects each investigator goes his or her own way with little or no communication with other investigators or even a feeling of responsibility to see that data are gathered so that the needs of this particular assignment will be fulfilled. What results is a series of inventories of varying scientific sophistication which are practically never useful for modeling or predictive purposes.

The determination of biological and ecological integrity is also hampered by the focus of attention on "pipe standards" rather than "receiving system standards." Thus, relatively little grant funding has been available from either governmental or private sources to develop methods for the quantification of the effects of pollution on biological integrity. One can obtain funding for purely "ivory tower" ecological research, but if one made the major thrust of this research the assessment of pollutional effects, it would almost certainly be disallowed by the more "ivory tower" funding agencies. Mission-oriented agencies have been focused on "pipe standards" rather than on "receiving system standards" and on chemical-physical measurements rather than biological measurements. Although it was in industry's enlightened self-interest to support the development of such methods, funding from this source has historically been trivial.

In addition to these difficulties, until recently there were relatively few scientific outlets for publication of such investigations and very little academic prestige attached to their production. As a result, most of the work was carried out by consulting firms or academic institutions which produced proprietary mimeographed reports of their investigations. Thus, what little knowledge was generated in this field generally has been given very limited distribution in the form of proprietary reports which were rarely subjected to peer review and certainly did not go through the rigorous scrutiny that occurs when publication is through the usual academic outlets and subjected to printed rebuttal, et cetera.

A brief statement of my own view of the conditions which produced our present state of disarray regarding the quantification of biological integrity follows. We do not know, in any scientifically justifiable sense, the characteristics of aquatic ecosystems which are essential to the maintenance of biological integrity. We also know practically nothing about the relationship between the structural and functional characteristics of natural ecosystems. Such factors as spatial distribution of species and the factors which cause systems to oscillate both in structure and function are so poorly documented that it is difficult for us to say what is desirable and what is undesirable except in the grossest way. Furthermore, most of the systems in the continental U.S. and particularly that area east of the Mississippi River have been in many ways substantially affected by man-initiated activities such as deforestation, flood control, agricultural activities, and so forth. Therefore, most of the systems with which we must work are already disturbed to some degree.

However, all is not lost! Most of the ecosystems in England, for example, have been influenced by human activities for generations and yet we find them pleasing and acceptable. Other systems such as the Thames River were once sufficiently degraded to be an objectionable nuisance and have been restored by planned reclamation efforts to a condition which, if not comparable to the primitive or original condition, is nevertheless more pleasing and more acceptable as well as more useful to us socially.

We are also able to estimate with reasonable precision the concentrations of toxicants which permit survival and adequate function of aquatic organisms which we consider important or representative. Given our present situation with a proliferation of chemical materials and a paucity of information on their toxicity, we might use as a reasonable working hypothesis that concentrations of chemicals and other potentially toxic materials permitting survival coupled with adequate growth and reproductive success will also permit the organisms to function reasonably well in other important respects.

We also know that aquatic communities subjected to pollutional stress will undergo structural alterations of a predictable nature. We know that the number of species will be reduced and that the number of individuals in certain species may increase. We can assess, on a site specific basis, such important behavioral characteristics as the temperature preference and avoidance of fish. Although the methodology for the assessment of biological integrity certainly could be markedly improved, the use of the methodologies in which we have confidence and a long history of effectiveness is still miniscule.

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DISCUSSION

Comment: I've been fascinated with some of the work that you and your students have been doing using individual fish or small groups of fish to monitor changes in water quality. At the outset of your talk, you indicated that you didn't feel that single species could be used in a measure of biological integrity. Would you comment on that?

Mr. Cairns: You're perfectly right. I probably should have covered this in the talk and it is in the paper. Single species are not a good index of biological integrity but can be used in an "early warning" system. The purpose of the single species in our inplant monitoring systems is to give an early warning of toxicity before the wastes actually reach the receiving system. This has some advantages but also all of the disadvantages of a "representative" species.

If an in-plant monitoring system showed biologically deleterious changes in the plant waste before it reaches the river, then you would have several options: (1) Shunt the waste to a holding pond; (2) recycle the waste for further treatment; (3) scale down the plant operations until the signal disappears.

This in-plant information would have to be correlated to the response of the biological community in the receiving system itself. That is the way it should be used—not as a single species index standing alone.

Perhaps I'm being too much the devil's advocate in not relying on individual species. However, it is dangerous to over-rely on individual species. There is a role for them in in-plant monitoring systems, but never without being coupled to some monitoring system based on community structure in the receiving system itself. Our in-plant and in-ecosystem monitoring units are designed to be coupled together.

Comment: I was intrigued by your reference to using natural systems to accomplish tertiary treatment, as opposed to manmade systems to accomplish the less costly primary and secondary. Were you referring only to non-toxic kinds of materials as opposed to relying on nature to handle say, chlorinated hydrocarbons and heavy metals? Also, do you feel that there is sufficient natural assimilative capacity to handle all of man's municipal and industrial wastes beyond secondary treatment?

Mr. Cairns: There are several points to be made here. One is that it may not be possible for many systems to handle all of man's wastes because they are too small and/or they are already overloaded. One would have to decide on a site specific basis.

As you point out, there are certain kinds of compounds which are neither degraded nor dispersed; these may undergo biological magnification and for these there is zero assimilative capacity. However, for most wastes many ecosystems have some assimilative capacity despite the zero discharge philosophy which assumes that one can forever contain the environmental effects of an industrial society. This is not possible because very, very advanced treatment requires energy, chemicals, and equipment.

The production of these will, ultimately, produce environmental effects somewhere. They will just be displaced from one site to another. I was trying to address that point in a very simplified fashion, but if there's no such thing as a natural assimilative capacity, we're in real trouble! That would mean the end of industrial society. So even though we have mostly anecdotal evidence and intuitive feelings that there is such a thing as assimilative capacity, we have no other choice but to assume nature can assimilate certain types of wastes and transform them. But if we don't define assimilative capacity more vigorously the assimilative capacity is exceeded and then the ecosystems will collapse.

We might set arbitrary standards for waste discharges based on general rules, and they might actually work and protect ecosystems, but then we'll be underusing the assimilative capacity most of the time. We will not be making full beneficial, non-degrading use of assimilative capacity, let's say, 364 days out of the year. Without information feedbacks about the condition of the receiving system good quality control is unlikely. My feeling is that the money spent on getting feedback of information about the response and condition of the receiving system would be more than offset by the savings in waste treatment if one linked the discharge operation to the receiving capacity.

So, what we should be trying to do, really, is mesh two dissimilar systems. One is an industrial system operating under, more or less, the market system and the ecosystem which is "controlled" by environmental variables. We should be trying to get these two systems working together in some optimal way for society's benefit; I feel that we can do much better than we're now doing in that respect.

Comment: The Agency's Water Quality Standards program, traditionally, has been based on providing levels for chemical parameters which is to provide protection for instream water quality and biota.

Do you feel that is an adequate system to provide protection for aquatic communities, or do you think instream levels of chemical parameters must be augmented by biological monitoring, some type of biological monitoring requirement in the water quality standards themselves?

Mr. Cairns: If you develop one standard for the whole country, this fails to consider how different ecosystems are in various regions and that these differences influence toxicity.

Another important factor is that one shouldn't regulate these toxicants individually and in isolation from others. We aren't exposed to toxic insults one at a time and as organisms we respond to the collective insults to our bodies (the smoke, contaminants in the water, and so forth). So do aquatic organisms and other organisms. The attempts to regulate one stress in isolation from the effects of other stresses won't work.

We should take advantage of the fact that natural communities summarize and integrate all these insults and give us a cumulative response. I agree that we should make some attempts to estimate the thresholds of toxicity for individual compounds since this is useful information. It is good predictive information, but ultimately we must go to the system itself and study the cumulative impact and the integrated response. I can't see any other way out.

Comment: I'd like to see you follow along a little in that. Can you give us an idea of what level of training would be necessary to have the toxic people go out and do this sampling that you suggested and, obviously, the money and manpower required to do that on a national scale?

Mr. Cairns: In today's dollars or tomorrow's dollars?

Comment: Take your pick.

Mr. Cairns: There are simple bioassays like the ORSANCO 24-hour bioassay. The round robins showed that it could be used by people who had no prior training. The sequential comparison diversity index can be used by people with no formal taxonomic training.

For such tests one can take high school graduates and, in one week, one can train them to produce useful, statistically reliable results.

These technical people do better with the simple tests than Ph.D's because the Ph.D's get bored and the other people don't. For simple tests can use relatively untrained people, as long as they are discriminating and dependable workers.

The aquatic ecology group at Virginia Tech has just developed a test using *Daphnia* for the American Petroleum Institute which can be run quite well by people who have had other types of formal training (such as chemical engineering, sanitary engineering) but with no training in bioassay methods.

To cut costs we are going to automation in our own lab. We have a unit using laser holograms which will probably identify 8,000 diatoms, when it's working at full effectiveness, in less than 10 minutes. Biologists, in general, have not taken advantage of computer technology and other types of technology to cut costs in water quality assessment.

The in-plant system that Dexter just mentioned is being installed at the Celanese Plant, Narrows, Va., with the Manufacturing Chemists Association. The capital investment for that, if you already have a mini-computer, will be about $26,000 and it can be operated by a high school graduate. The laser system would cost about $156,000 in today's dollars, but could be used by many plants. The cost per analysis using automation will be relatively slight.

For a complete stream survey using nine taxonomists, my guess would be $4,000 to $8,000 per station for each examination. The "complete physical" surveys are very expensive. Functional measurements could also be expensive, except for the simple rate processes.

Comment: We're not only an industrial society, we're largely an urban society. There're a great number of these sections doing area-wide studies, an awful lot of them underway around St. Paul.

Many of the streams in urban areas are subject to large sediment loads, lots of scouring due to runoff a priori. Can we establish biological integrity with those kinds of streams, because if the locals are going to be added, they will have to choose between alternatives. They will need some measure of what thought, or the possible results to those alternatives when you're talking about control of runoff, perhaps treatment and other kinds of disposal that are in place now.

Do we have to take the physical scenarios of physical alterations for improvements? The same thing on chemical, do we equate those and then try to come up with the biological scenarios and possible results, and then just lay them out and take your chances?

Mr. Cairns: I don't know if there is anybody here that can answer your question.

My guess would be, if I understood your question correctly, that in systems like the Ohio we'll never find out the original condition, so we must set standards (chemical, physical, and biological) that are satisfactory to us as a society. The various scenarios with cost-benefit analyses can be given to the general public or other decisionmakers for final choice.

If you look at the ORSANCO reports it is evident that the general water quality trend has been toward improvement of chemical-physical characteristics and they are getting more stable and more predictable. This resulted from an implemented regional scenario.

Comment: I was talking about any of the much smaller water bodies that are around the country. We only have a few very large systems. In many areas the streams are much smaller, they're creeks, and bodies down to that size.

A great deal of money could be spent, but we have to base some decision on biological integrity.

Mr. Cairns: We're studying the South River with DuPont in Virginia. It's a very small stream with very heavy waste loading. I believe we can evaluate biological integrity for this stream and develop practical management programs to either improve or maintain present condition of integrity.

It would not be cost effective to restore that stream to its original condition, but I think it is reasonable to restore the stream to some more acceptable condition before it joins the Middle and North Rivers to form the South Fork of the Shenandoah River. There are acceptable means of determining biological integrity and developing management plans. Implementing then is another matter.